Wood ash use in forestry – a review of the environmental impacts

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The use of wood fuel for energy production in the UK is set to increase in the near future as part of a government commitment to increase renewable sources to 10 per cent by 2010. The ash generated as a by-product of combustion, whether for heat or power generation, has potential use as a fertilizer in forest systems. This review assesses the available information on factors affecting the quality of the ash and environmental implications arising from its application. The key determinants of wood ash chemistry are the tree species combusted, the nature of the burn process and the conditions at the application site. Wood ash from hardwood species produces higher levels of macronutrients in their ash than conifers, and the silica content is frequently lower. A furnace temperature between 500 and 900°C is critical to the retention of nutrients, particularly potassium, and determines the concentrations of potentially toxic metals including aluminium in the ash. Fly ash, the lightest component that accumulates in the flue system, can contain high concentrations of cadmium, copper, chromium, lead and arsenic and this ash should not be used as fertilizer. The form of the ash at application is important, with loose ash releasing Ca, K and Na more rapidly than granulated ash. Heavy metal, radionuclide and dioxin contamination of wood ash-based fertilizers is minimal and unlikely to affect ecosystem function. The effects of wood ash are primarily governed by application rate and soil type. The benefits are maximized at low dose rates, with possible toxicity from applications in excess of 10 t ha^{-1}. For most forest sites, a single wood ash application per rotation could replace all the nutrients lost after whole-tree harvesting (excepting N). Long-lasting positive effects on tree growth have been observed on shallow peats, in which the humus is slowly mineralized in response to elevated pH and increased nutrient availability. In contrast, wood ash application to podzols is only effective in enhancing tree growth when nitrogen availability is non-limiting. To date, published research of wood ash effects on trees growing in clays and loams is minimal. A lag time for positive tree responses to wood ash application is often observed, and may be the result of phosphorous limitation at higher soil pH. The greatest reported adverse ecological effects are to acidophilic ecosystems, particularly the constituent bryophyte, soil bacteria and ectomycorrhizal communities.

Introduction

The UK Government is committed to reducing CO_2 emissions under the Kyoto agreement by at least 12.5 per cent from the 1990 rates through

the period 2008–2012, and has a policy of promoting renewable energy sources as part of the solution (DefraUK, 2003). The Renewables Obligation introduced in April 2002 requires electricity

suppliers to obtain 10 per cent of this supply from renewable energy sources by 2010, and aspirational targets of 20 per cent green energy were set in the Energy White Paper in 2003. Further targets to replace fossil fuels by biofuels up to a target of 5 per cent by 2020 have highlighted the potential importance of the available wood fuel resource in the UK, which has recently been estimated at 3.1 million tons per year (oven dry), in the absence of competing markets (McKay, 2003). This would potentially come from small round wood (7–14 cm diameter), sawmill co-product, utilities and roadside clearance, lop and top from harvesting operations and short rotation coppice planted on agricultural land. Estimates of dry materials given in the Biomass Task Force report to Government (DefraUK, 2005) estimate current forestry waste in the UK at 1.5 million tons, industry waste wood at 3 million tons and energy crops, including short rotation coppice, at 0.2–0.4 million tons. Since the commercial burning of wood generates up to 1 per cent ash by weight, a significant waste by-product is created, requiring costly disposal or beneficial reuse as forest fertilizer, as practised in Scandinavia (Swedish National Board of Forestry, 2002). To date, wood ash has not been created on a commercial scale in the UK, but expected ash production from large energy plants, such as that commissioned on Teesside (30MW power station) by SembCorp Utilities (UK), will be some 3000 tons annually, from 300 000 tons of wood. Ash is successfully used elsewhere on agricultural land, for other purposes such as sewage amendment, scrubber systems, cement products (Greene, 1988) and for road building in both Scandinavia and the USA.

Although ash recycling to forests seems a good ecological solution, there are a number of potential problems associated with its use in forest systems, which are more complex than those in agriculture. Agricultural soils are maintained as near as possible to neutral pH during cultivation, and fertilizer amendments are disked into the topsoil for immediate effect over short time periods. Forest soils by contrast may be poorer, thinner and often more acidic, particularly under conifers – ash addition may then increase nitrogen levels in soil leachate. The longer life of the crop complicates the treatment of the site, and the structure of the crop affects the physical delivery of soil amendments. This review summarizes the current knowledge of best practice for using wood ash as fertilizer to forestry systems, its composition and its effects on the sustainability and biodiversity of soils.

Background to wood ash research

Research into the composition and use of wood ash comes from four main source countries – USA, Finland, Sweden and Denmark. In the USA, wood ash is derived from paper industry waste and power generation and nationally 90 per cent of this goes to landfill. However, in the north-east states only 15 per cent is landfilled, as the remaining 80 per cent is land applied and 5 per cent co-composted with sewage sludge. This practice has reduced the costs of disposal for the producing companies by up to 66 per cent in Maine and New Hampshire (Greene, 1988; Campbell, 1990; Vance, 1996).

In Finland, wood ash has been used as a soil ameliorant for second-rotation conifer stands on drained peats, on sites monitored since 1935 (Hakkila, 1989; Korpilahti *et al.*, 1999). Research into the restoration of cut-over peat using ash as an ameliorant to change pH and restore biodiversity is underway (Näsi *et al.*, 2005). In Sweden, ash is already produced in large quantities from energy generation and research on recycling ash to forest sites on peats and podzols was instigated in the 1970s (Högbom and Nohrstedt, 2001). Recently, research to restore acidified soils in the south of the country has explored the use of both wood ash and lime (Lundström *et al.*, 2003). In Denmark, ash is produced from community energy projects using mixed organic fuels such as straw, woodchip, green waste and tree thinnings. This has resulted in a mixed quality ash of variable chemical content with some high levels of heavy metals and dioxins. As a consequence, 2500 tons of ash per year is disposed to landfill (Serup, 1999; Møller and Ingerslev, 2001).

Composition of wood ash

Source of the material

The nature of the wood fuel strongly influences the composition of the wood ash (Table 1). Residues

Table 1: Element concentration in wood ash from various sources – % dry weight

	Wood residue/bark ash*		Paper mill ash†	Softwood ash‡		Hardwood ash‡	
	Median	Range	Median values	Stem	Bark	Stem	Bark
Ca	13.2	7.4–33.1	16.6	22.4	28.5	19	27.1
Fe	1.51	0.3–2.1	0.51	0.8	0.2	0.5	0.6
K	2.93	1.7–4.2	2.57	12.4	9.8	20.4	12.2
Mg	1.47	0.7–2.2	1.07	4.3	2.8	3.6	2.2
Mn	0.67	0.3–1.3	0.32	2.9	1.7	0.8	0.6
Na	0.24	0.2–0.5	0.1	–	–	–	–
P	0.79	0.3–1.4	0.39	2.4	2.8	4.2	3.4
S	0.56	0.4–0.7	0.02	2.3	1.2	2.1	1.1
Al	2	1.5–3.2	0.91	–	–	–	–
C	–	–	25.5	–	–	–	–
pH	12.7	11.7–13.1	12.4	–	–	–	–
N	Rarely reported <0.1%						

Data from *Campbell (1990), †Muse (1993) and ‡Hakkila (1989).

from the paper industry burn to produce a different composition of ash to that from bark-burning boilers or tree harvesting residue burns (Naylor and Schmidt, 1989; Campbell, 1990; Muse and Mitchell, 1995). Someshwar (1996) and Hakkila (1989) both showed that this mixed ('hogged') fuel, composed of paper-processing residues and waste wood results in very variable ash chemistry. Elemental composition varies with the type of plant tissue included in the wood fuel. Ash derived from branch and root wood is richer in many elements than stemwood, while that derived from bark and foliage have concentrations between five and 10 times greater than from stemwood (Hakkila, 1989; Werkelin *et al.*, 2005). High values of Ca and Si are present in bark ash, and Mn, Al and Si are concentrated in the mixed stemwood. Bark thus produces more ash than stemwood (6 per cent compared with 0.25 per cent). Ash production from beech, birch and pine species is lower than from spruce or aspen species (Serup, 1999; Werkelin *et al.*, 2005).

The composition and quantity of wood ash is also dependent on the tree species burnt to produce the wood fuel, as summarized from various sources in Table 2. Variation in published values between species is confounded by variation in soil type and climate. In general, hardwood ash contains more K and P than softwoods but less Ca and Si. There are notable differences even in the same genus of pines (Naylor and Schmidt, 1986), but less variation among oak and poplar species. The poplar species produce much less Ca compared with other broadleaves: two-thirds of that from oak and half that from birch and maple species.

Technological effects on ash composition

A key factor in final ash composition is the temperature of the burn and whether the furnace creates separate fly ash and bottom ash. Fly ash is the lightest waste component from the burn (~200 μm), which is deposited on the inside of the boiler and ventilation systems and often contains higher levels of dioxins and heavy metals than the bottom ash.

American authors have noted differences in the ash quality and quantity, from domestic wood stoves and commercial boilers, which commonly fire at below 1200ºC and above 2000ºC, respectively.

Etiégni and Campbell (1991) observed a 45 per cent reduction in ash generation from lodgepole pine from burns between 500ºC up to 1400ºC and Misra *et al.* (1993) demonstrated that this effect of temperature is species dependent. Ash mass loss from five tree species fell by between 23 and 48 per cent over the temperature range of 650 to 1300ºC. Figure 1 shows the typical changes in element concentration (relative to calcium) with increasing burn temperature for aspen. In pine, aspen and white oak, the volatilization

Table 2: Element concentrations in wood ash from specified tree types (mg kg^{-1})

Macro element	Al	Ca	Fe	K	Mg	Mn	Na	P	S	Si
Conifers										
Pinus banksiana	33.3	387	35.0	22.5	33.2	39.0	23.0	12.2	10.4	74.8
Pinus sylvestris (W)	1–18	600	3–15	300	120	70.0	3–22	30.0	NG	NG
Picea abies (W)	NG	700	NG	300	90.0	90.0	NG	20.0	NG	NG
Pinus sp	4.7	290	5.8	162.5	70.3	40.4	0.6	8.4	10.7	ND
Tsuga heterophylla	11.1	421	9.1	25.3	79.0	19.0	8.2	9.2	5.6	46.7
Broadleaves										
Betula sp	0.0	466	20.3	36.3	25.3	47.0	9.6	12.6	12.8	14.0
Betula pubescens (W)	3.0	500	7.0	400	90.0	90.0	7.0	40.0	100	90.0
Acer sp	20.1	402	11.9	31.9	117.0	27.0	16.3	4.8	5.6	46.3
Populus tremuloides	1.4	212	2.6	112.5	35.5	1.4	0.6	11.8	7.0	1.1
Populus sp	3.5	257	3.2	79.3	90.9	4.5	23.0	9.5	10.2	ND
Quercus rubra	6.8	366	NM	60.8	52.0	14.9	0.8	15.6	18.0	ND
Quercus (white oak)	ND	314	0.9	102.5	75.7	1.4	ND	5.6	12.1	1.3

Data from Werkelin *et al*. (W) derives from a low temperature burn of 575°C.
Data from US sources: Someshwar (1996), Misra *et al*. (1993), Mingle and Boubel (1968) and Werkelin *et al*. (2005).
ND = not detected, NM = not measured, NG = not given.

appeared to occur in two steps. This was attributed to decomposition of calcium carbonates between 650 and 900°C, and the carbonation of K to K_2CO_3 at higher temperature.

Unburned carbon concentrations in commercial boilers in eastern USA commonly form 7–50 per cent (mean 26 per cent) of the ash (Someshwar, 1996). Similar values have been reported for conventional grate boilers in Sweden, but circulating fluidized bed (CFB) boilers are more efficient and allow fly ash to be cleared out for potential retrieval of carbon (Tollin, 2000). The lowest ash volumes quoted in American literature, under 5 per cent, come from boilers with inclined, sliding pinhole grates (Etiégni *et al*., 1991). Small-scale boilers in Britain are currently very efficient and produce less than 1 per cent ash by weight. Apart from indicating an inefficient burning process and creating unnecessary bulk, a carbon content over 20 per cent hinders the process of ash granulation and chemical hardening (Etiégni *et al*., 1991) and adds extra cost to handling, transport and spreading operations (Wilhoit and Quingyue, 1996).

Volatilization of K and S occurs above 8–900°C and 1000–1200°C respectively, resulting in losses of 63 and 90 per cent K and 7–55 per cent S (Naylor and Schmidt, 1986). Other authors showed that S, B and Cu weights decrease with burn temperature, but this is less clear for Na and Zn. Misra *et al*. (1993) reported that the overall concentrations of Mg, Zn, Mn, P and Si show little change with temperature relative to Ca, which is assumed to be constant. However, this is variable according to tree species, as shown in Figure 1 for aspen, where Si values only stabilize at over 800°C. But K and B values fall at the same temperature. Etiégni and Campbell (1991) suggest that the decline in carbonate content from 63 to 51 per cent as temperature increases to 1093°C is related to the formation of carbonates of Ca and K at low temperatures (<900°C). Above 1300°C, both calcium and magnesium oxides and other metal compounds are generated. Ash composition is much modified by the presence or absence of Si, Mn, Fe or Al, all of which may also form oxides, which can combine with more alkali compounds to form sintered ceramic-like deposits in the ash (Misra *et al*., 1993). These elements have synergistic effects on one another at burn temperatures above 900°C, when molten potassium carbonate and sulphate adhere to cooler metal surfaces and trap other particles, such as the oxides of Ca and Mg. Highest levels of macronutrients will be retained therefore in a burn temperature between 500 and 800°C.

Someshwar (1996) has summarized data from other American authors on trace and heavy metal
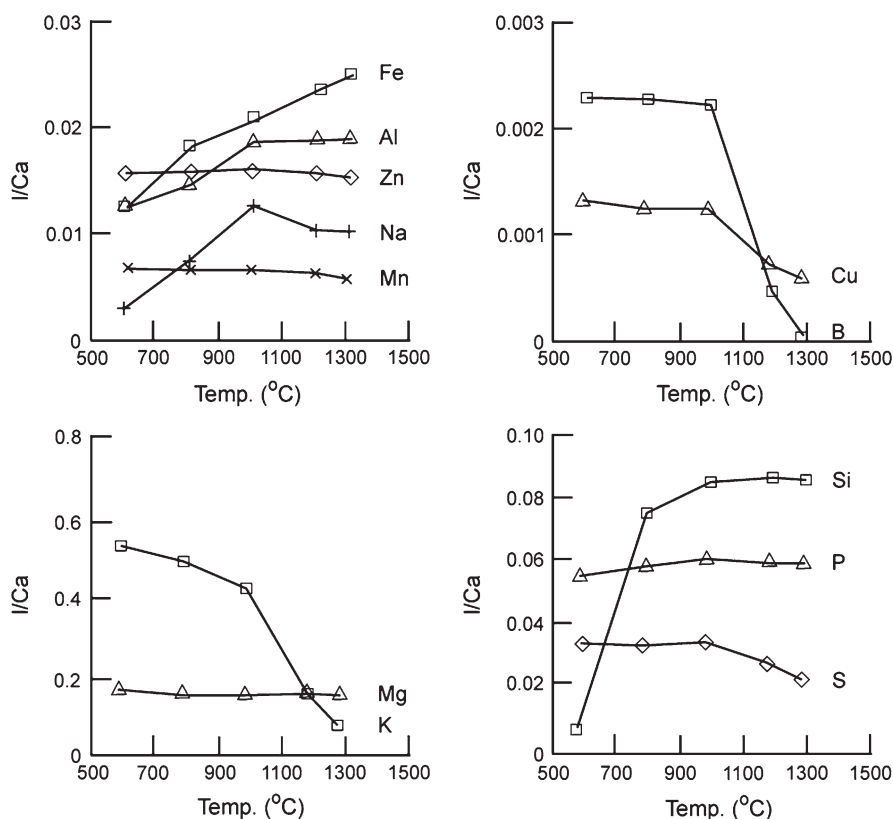
Figure 1. Variation of normalized concentrations of different elements in aspen ash with burn temperature. (Redrawn from Misra *et al*., 1993). Fe = Iron; Mn = Manganese; B = Barium; Cu = Copper; Si = Silica; Na = Sodium; Zn = Zinc; K = Potassium; Mg = Magnesium; P = Phosphorous; Al = Aluminium; S = Sulphur; I/Ca = element concentrations relative to Calcium.

production, with ash analyses from 26 fuel wood combustion boilers (Table 3). Zn and Mn concentrations are very variable, but always high (median levels of 329 and 3485 mg kg^{-1}, respectively). These levels are confirmed in other studies by Meiwes (1995) and Välttillä *et al*. (1994). The latter assessed the effects of paper mill sludge addition to three different wood ash burners – of these, the grate boiler burn produced the highest levels of Cr, Cu, Ni and Pb from identical mixes, and the CFB boiler produced the least.

Boilers that separate fly ash from bottom ash provide a better source of wood ash for use as fertilizer. This is because vaporization of metals in the combustion process (particularly at higher temperatures) is followed by condensation on cooler surfaces away from the heating zone,

resulting in high heavy metal concentrations in fly ash (Hakkila, 1989; see Table 4). Zn is the exception, and is generally higher in bottom ash. Nordin *et al*. (2005) have recently refined high temperature techniques to volatilize these metals and remove them in extracted gases from both wood ash and sludge residue. Further information is available from IEA Bioenergy, (2004), which summarizes existing ash chemical data from European countries for various boilers and sources of biomass.

Polyaromatic hydrocarbon (PAHs) occurrence in wood ash are generally the less toxic compounds, with naphthalene the most common (Diebel *et al*., 1992). Despite the large inputs of polychlorinated biphenyls (PCBs) into the natural environment recorded in western countries between the 1950

Table 3: Concentration of heavy metals in wood boiler ash (mg kg^{-1})

Element	Wood boilers* (median values)	Bottom ash† (range)	Fly ash† (range)
Hg	<3	<0.4	0–1
Se	<3	–	–
Cd	<25	0.4–0.7	6–40
Co	–	0–7	2–300
Mo	<50	–	–
As	–	0.2–3	1–60
Cr	–	>60	40–250
Ni	–	40–250	20–100
Pb	<110	15–60	40–10^3
Cu	–	15–300	~200
V	–	10–120	20–30
Zn	>300	15–103	40–700
Mn	–	(2–5.5)103	(6–9)10^3

*Someshwar (1996) and †Hakkila (1989).

Table 4: Element concentrations of hardened and granulated ash – % dry weight

Ash type	Ca	Mg	K	Na	P	S	Zn	SiO$_2$
Loose	21.1	2.0	3.2	1.1	1.2	1.2	0.1	11.8
Crushed	18.2	2.0	1.4	1.2	0.5	2.1	0.1	26.2
Granulated	16.4	1.6	4.0	0.9	2.2	2.2	0.1	22.0

Data source: Kellner and Weibull (1998) and Nilsson and Lundin (1996).

and 1970 (Harrad *et al*., 1994; Pichler *et al*., 1996), PCB uptake by trees appears to be limited (Bundt *et al*., 2001). Consequently, wood ash generated from forestry arisings contains no traces of PCB in any fraction – this is also the case for chlorobenzines and chlorophenols (Someshwar, 1996). Dioxins and Furans (PCDD/Fs) are immobilized in wood ash and are unlikely to leach out of wood ash due to its absorbent nature. The PCDD/F contents of both 'domestic' grate ash and 'commercial' wood boilers are negligible (Kuykendal *et al*., 1989; Diebel *et al*., 1992) although they are present in chimney soot (Someshwar, 1996). However, ash produced during the combustion of sea salt laden wood, can result in high levels of PCDD/Fs and dioxin in the fly ash, but not in bottom ash (Luthe and Prahacs, 1993). Fly ash from European boilers generally has higher levels of PCDD/Fs compared with ash from inland USA (Pohlandt and Marutzky, 1994; Välttillä *et al*., 1994), which has been attributed to the higher chloride concentrations (>0.03 per cent) typical of wood grown in the oceanic climate of western Europe (Someshwar, 1996).

The physical form of the ash

A number of problems are associated with the application of loose ash. Firstly, loose ash is difficult to handle and presents health risks to operators from fine airborne particles (Hakkila, 1989). Secondly, loose ash is difficult to spread evenly, and necessitates slow delivery rates from spreaders (Wilhoit and Qingyue, 1996). Finally, loose ash has a greater potential for deleterious effects to ground vegetation, particularly to the cover and species of moss communities (Kellner and Weibull, 1998). Preparation methods commonly in use in Sweden produce 'crushed' or 'granulated ash'. In the former product, bottom ash is taken from the CFB boiler, moistened to 30 per cent water, allowed to harden for 4 weeks and then dried out – the 'self-hardening process' (Neilsson *et al*., 1996; Korpilahti *et al*., 1999). It is then crushed and sieved to achieve particles under 5 mm. Granulated ash is produced by mixing loose bottom ash from the CFB boiler with water and rolling it to form balls 4–20 mm in size. These

are then dried to a water content of <5 per cent (Kellner and Weibull, 1998). The three ash types have differing chemical composition with lower calcium and higher phosphorous content in the more processed products (Table 4).

In a series of laboratory experiments, Steenari *et al.* (1998) recorded nutrient losses during alternate wetting and drying cycles using both self-hardened and granulated ash. The small size self-hardened ash lost calcium and potassium content faster than the granulated CFB ash, which

retained its structure and 90 per cent of its weight after 2 years simulation time. By this point, most of the small self-hardened ash had dissolved (Figure 2). Eriksson (1998b) also tested loose and crushed self-hardened bottom ash (with separate fractions above and below 5 mm size) in a series of column experiments. Analysis of the ash remaining after the simulation of 5 years rainfall showed that all fine loose ash bags lost nutrients (K, Na, Cl and SO_4) rapidly, whereas the loss rate from the hardened ash was much slower. The
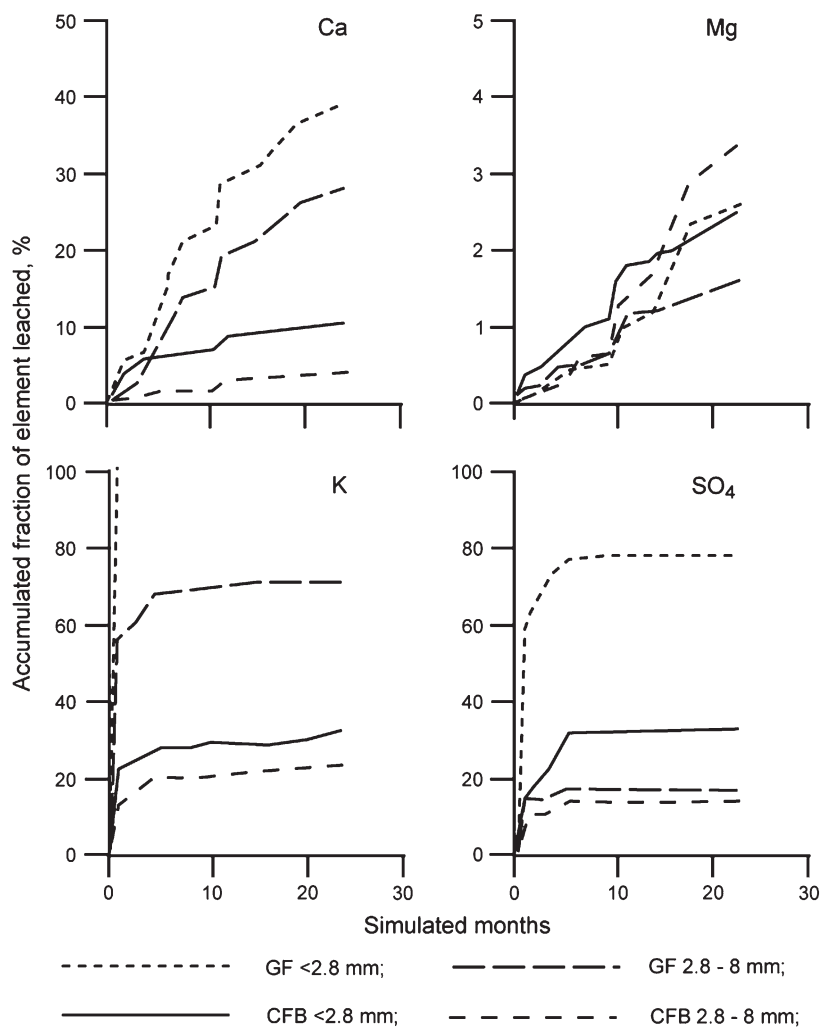

Figure 2. Accumulated fractions of elements leached during simulated weathering. (Redrawn from Steenari *et al.*, 1998). Ca = Calcium; Mg = Magnesium; K = Potassium; SO_4 = Sulphate; GF = grate-fired boiler, self-hardened ash; CFB = circulating fluidized bed boiler ash – granulated.

mobility of heavy metals was equivalent to that released in the field from pine needle litter decay on the forest floor (Laskowski and Berg, 1993).

In a short-term complementary field study, Eriksson (1998a) found that rises in pH, cation exchange capacity (CEC) and base saturation levels in a podzol after the application of ash granules remained positively correlated with the ash dose, which had varied between 1 and 6 t ha^{-1}. Steenari *et al.* (1998) recorded an initial, rapid loss of calcium from hardened ash, which stabilized within a few weeks. After a year in the field, loose ash had lost 50–60 per cent of the original potassium and sodium, whereas in the hardened ash calcite, gypsum and Ettringite had formed. In newly burnt ash, the calcium oxidizes through Ca$(OH)_2$ (Portlandite) to calcite ($CaCO_3$) in the presence of atmospheric moisture. This becomes further transformed on saturation into Ettringite ($Ca_6Al_2 (SO_4)_3(OH)_{12}$. $26H_2O$), with a reduced calcium leaching rate, reduced alkalinity and more moderate pH. Mobility of heavy metals in this processed ash is also reduced. A recent recovery study by Nieminen *et al.* (2005) confirmed this finding for heavy metals, particularly Pb, Ni and Cd. Release of nutrients from self-hardened and granulated wood ash was monitored 3 and 5 years after spreading onto forest soils – K, Na, B and S were easily removed from both types of ash, though less in the granulated product, but while concentrations of Ca, Mg, Zn and Cr decreased in the self-hardened ash, they remained constant or increased in the granulated ash. It is clear that granulated ash represents a superior formulation for application and is now strongly recommended by practitioners in Sweden (Swedish National Board of Forestry, 2002) and Finland (Nieminen *et al.*, 2005).

Water holding capacity

Etiégni *et al.* (1991) studied the structural changes to wood ash during wetting, and suggested that ash is essentially hydrophilic and absorbs water into the pores by capillary action simultaneously with chemical changes of hydration of oxides. Scanning electron microscopy of wood ash revealed many irregularly shaped inorganic particles with thin layers of crystalline structures which swelled to clusters of rosette crystals, resulting in a volume increase of 12.5 per cent after wetting. The probable compounds, determined by X-ray patterns, were calcite, portlandite and calcium silicate. Magdoff *et al.* (1984) as reported in Campbell (1990), have shown that, due to the alkaline conditions caused by the ash and its liquid retention properties, both herbicides and pesticides will be readily adsorbed by ash on the ground. Ash application and other spraying should be staggered to avoid the possible reduction in the effectiveness of applied pesticides and herbicides.

Alkalinity characteristics and dose rates

Generalized comparisons with commercial fertilizers such as ground limestone, hydrated lime and potash have been made to support the case for the use of wood ash in US agriculture (Naylor and Schmidt, 1986; Campbell, 1990; Someshwar, 1996). Nevertheless, in some states wood ash is classified as hazardous waste when its pH exceeds 12.5 (Washington), and in New York, Maine and New Hampshire there are significant regulatory restrictions to its use.

The alkalinity of ash or its equivalent neutralizing values (ENVs) is defined in the USA by the calcium carbonate equivalent expressed as percentage of standard limestone (equated to 100 per cent). The ENV's of hogged fuel ash can vary considerably, with a range from 13.2–92.4 per cent reported by Vance (1996). Hakkila (1989) published ENVs from various fuels from pure wood ash at 115 per cent (compared with standard lime) to bark ash mixes at 64 per cent, highlighting the role that bark may play in reducing the alkalinity of wood ash. The liming effect of ash addition on the soil pH is a function of both calcium and magnesium carbonates and the fineness of the material. Depending on the burn efficiency, wood ash may also include charcoal, which reduces its final neutralizing capacity. In Europe, the acid neutralizing capacity value is typically half that of carbonate and silicate lime (Meiwes, 1995). The most succinct comparison of wood ash and lime ENV comes from laboratory soil experimental work by Naylor and Schmidt (1986) in which the equivalence of ground limestone and wood ash was compared, using two soils common in north-east USA, Mardin and Burdett silt loams. After soil incubation of 60 days, with alternate wetting and drying

cycles, the pH change was consistent for lime and ash, at an equivalence of 47–50 per cent.

Wood ash effects on soil properties

pH changes

An indication of optimum wood ash application rates can be provided by comparison with guidelines for liming, which are calculated for forests using a target value of the base saturation and cation capacity of the soils and knowledge of the tree crop type. Andersson *et al.* (1995) made recommendations for forest soils in Sweden, targeting a humus layer of base saturation at 50 per cent, and a mineral soil base saturation at 20 per cent, equivalent to a lime requirement of 2–5 t ha⁻¹. Tree species such as beech (*Fagus sylvatica* L.), oak (*Quercus robur* L. and *Quercus petraea* (Matt.) Liebl.), Norway spruce (*Picea abies* L. Karst), Scots pine (*Pinus sylvestris* L.) and Douglas fir (*Pseudotsuga menziesii* Franco) typically grow best with base saturation rates of 30 per cent, whereas others, such as hornbeam (*Carpinus betulus* L.) and sycamore (*Acer pseudoplanatus* L), prefer levels in excess of 50 per cent. However, deeper and more acidified soils might need up to 15 or 20 t ha⁻¹ of lime to reach this base saturation (Meiwes, 1995), although application rates this high would rarely be used, due to negative effects on existing ground vegetation. As already outlined, the delivery rate of calcium to soils is dependent on the initial form of the ash, with loose ash potentially causing a transient fast rise in pH in the soil (Kahl *et al.*, 1996). However, other long-term field trials indicate a sustained pH rise over several years after ash application – 5 or 6 years in the studies of Jacobson *et al.* (2004) for the humus layer of a podzolic soil dosed with 3.6 and 9 t ha⁻¹ and Fransman and Nihlgård (1995) detected a sustained rise in runoff water pH over the same time period from a site in Central Sweden dosed with 2.2 t ha⁻¹ wood ash. In Finland, Saarsalmi *et al.* (2001) resampled a pine and Norway spruce site given an ash dose of 3 t ha⁻¹ 16 years previously, and detected a pH rise for both humic and mineral topsoil layers under both canopies, most significant at the wetter sites. Elevated soil pH in the humus layer was maintained at a value of 0.6–1.0 pH units higher than control forest soils – the greatest effects on CEC

were however at the driest site with lowest initial pH. The mineral soil layers (below 10 cm depth) had showed very little change at 7 years after ash application, but after 16 years, the increased pH values at depth indicated a slow downward transfer of activity from the topsoil over time.

Nutrient additions by wood ash

Measurements of CEC quoted in Etiégni *et al.* (1991) give a value from the literature for wood ash of 2.7 meq/100 g. This ranks alongside inactive clay minerals such as kaolinite.

Naylor and Schmidt (1986) have equated wood ash fertilization effects to commercial fertilizers on the basis of the N, P (P_2O_5) and K (K_2O) concentrations. For a commercial wood boiler, this would be similar to a 0 : 1 : 3 (N : P : K) fertilizer, but from a domestic wood stove (lower temperature burn), the ash has a higher proportion of K and would be equivalent to a 0 : 3 : 14 fertilizer. These authors showed that the availability of K was directly related to the amount added to the soil. Only a proportion of the K applied as boiler ash (18–35 per cent) or wood stove ash (51 per cent) is available to plant uptake as a result of immobilization. This compares to 65–70 per cent availability from commercial potash fertilizer. K dissolution rate and availability are also dependent on soil pH (Naylor and Schmidt, 1986; Erich, 1991; Ohno, 1992). P supply from wood ash is also lower than from commercial fertilizers (28–70 per cent) and availability dependent on soil type (Erich and Ohno, 1992). It was concluded that the variation was due to pH buffering capacity of the soils, with P availability at a maximum for soil of pH 6.0–7.0, decreasing at pH over 8.0 (Etiégni *et al.*, 1991). There is also evidence that P may be immobilized as iron and aluminium phosphates in acidic soils (Ohno, 1992). Recovery of P 5 years after ashing from two soils of different pH in Central Sweden were four times greater in the more acid soil, indicating less uptake by the trees and/or less leaching (Jacobson *et al.*, 2004). Clarholm (1994, 1998) suggested that P in the ash is not in a water-soluble form, and by tracking the movement of 32P through both humus and fine roots of the trees, it appears that uptake was not related to tree demand, but was negatively related to the P/C ratio of the

microbial mass around the roots. The author suggested that the levels of P in the biota, 5 years after ash treatment, had significantly increased and would subsequently feed into increased P uptake by the trees.

Vance (1996) has suggested that one single application of wood ash at 10 t ha^{-1} could replace nutrient losses from whole-tree harvesting sites, along with additional N amendments to create balanced input. Wood ash application generally has little direct impact on N availability due to its low levels in the raw material (<0.1 per cent). However, N availability often increases indirectly as a result of ash application due to a rise in soil pH and consequent N mineralization. This has become an important concern at sites in Scandinavia, for podzolic and peat soils (Hogbom et al., 2001a). Increased N mineralization may be related to both increased soil microbial biomass and shifts in the ectomycorrhizal (EM) community (Taylor and Finlay, 2003), though other sites have shown little change 4 years after ash treatment at low levels (Lundström et al., 2003). Most research shows that the initial N status of the soil, its pH and the ash dose rate are all critical to potential increased N leaching rates.

Heavy metal contamination

In the UK, the Interdepartmental Committee on the Redevelopment of Contaminated Land (ICRCL, 1987) trigger concentrations of metal in contaminated soils have been the guide to acceptable threshold levels – those posing a hazard to human health, such as As, Cd, Cr, Pb, Hg and Se, and those that may be phytotoxic, but are not normally hazardous to humans, such as B, Cu, Ni and Zn (ICRCL, 1987). These trigger values have been updated (2003/5) with soil screening values, which have been proposed as a basis for rapid assessment of possible contamination across all land uses (DefraUK, 2003). At present, PNEC$_{soil}$ values (predicted no effect concentration), which will be accepted on a European wide basis, have been determined only for Ni (0.8 mg kg^{-1} of dry soil) and Cd (2.3 mg kg^{-1}). These are lower than the old ICRCL values in Table 5 and in general, European regulations limiting maximum metal concentrations in soils are more stringent than those in the US.

The variability of Zn, Ni and Cu in bottom ash poses a risk of exceeding permissible levels if ash is applied in large quantities, but typical Cr, Cd, As, Hg and Pb contents should present no risk, as long as fly ash is not used. Table 5, compiled from various sources, summarizes the data. The heavy metal concentration in agricultural fertilizers frequently exceeds those from bottom wood boiler ash.

Vance (1996) and Campbell (1990) have assessed the loading limits for heavy metal levels set in the USA for sewage sludge application

Table 5: Soil limits for heavy metals from UK and European regulations (mg kg^{-1})

	ICRCL (UK) soil 'trigger' concentration limit for open space	Sludge treatment (maximum soil limits)		Common fertilizer constituents concentration*			Bottom ash concentration variation[†]
		European Union	UK	Phosphate	Potassium	Farmyard	
As	40	–	–	–	–	–	0.2–3.0
Cd	15[‡]	1–3	3	9–30	7–15	0.3–1.8	0.4–0.7
Cr(T)	1000	100–150	400	1000	1000	12	40–250
Cu	50	50–140	135	–	–	–	15–300
Hg	20	1–1.5	1	<50 ng kg^{-1}	–	100 ng kg^{-1}	0–1
Ni(T)	70[‡]	30–75	75	300	80	17	40–250
Pb	2000	50–300	300	–	–	–	15–60
Zn	300	150–300	300	5–1450	–	15–250	15–1000

Data from *Alloway (1995) and [†]Hakkila (1989).
[‡] See text for proposed soil screening values (DefraUK, 2004).

(USEPA, 1993). They concluded that applications of ash up to 10 t ha^{-1} from ordinary boilers result in heavy metal soil levels still two orders of magnitude lower than the USEPA advised loading. They also suggest that changes brought about by added K and calcium carbonate effects on soil pH are likely to be more limiting to ash use on soils than heavy metal contamination. The individual adsorption and mobility characteristics of heavy metals in soil are obviously critical to understanding potential dangers to human health through agricultural crops (Alloway, 1995), though not so vital in forestry. Published guidelines in Britain (MAFF, 1993) are based on soil pH range (5–7.0 units), over which the adsorption of heavy metals in soil changes rapidly. The Swedish recommended limits for heavy metals in wood ash are given in Table 6.

The effects of cadmium on ecosystems are of particular concern, and it is one of four metals for which critical loads have been set by UN Economic Commission for Europe (Hettelingh *et al.*, 2002). Alloway (1995) has shown that pH, organic matter and hydrous oxide content are the key factors controlling Cd adsorption into soils. Wood ash with high Cd compounds has reduced risk of this metal leaching, as it bonds with hydrous Mn oxides, through a wide range of soil pH's (Mn is plentiful in both bottom and fly ash) and with hydrous iron oxides (common in many mineral forest soils).

The concentrations of Zinc recorded in bottom ash seem high compared with other metals, but the range is similar to those measured in commercial fertilizers and about half those of sewage sludge (Kiekens, 1995). In the European Union, there is a maximum allowable loading to soils of 550 kg ha^{-1} of Zn (assuming an average soil background content of 80 mg kg^{-1} Zn). Natural levels of chromium and nickel in the soil are determined by site geology – McGrath and Loveland (1992) have measured the highest levels in coarse loamy, sandy and peaty soils. They suggest that levels of Cr or Ni in fertilizers, sewage sludge or other soil amendments are unlikely to cause a build up of these metals in the soil above naturally occurring levels.

Wood ash contains large amounts of manganese, particularly when derived from conifers (Hakkila, 1989). However, it seems unlikely that addition of wood ash to temperate soils would result in any toxicity (Smith and Patterson, 1995). Mn does not appear as a hazardous metal in any

of the new soil screening values, the ICRCL tables, or those for the USA. A full review of heavy metal potential toxicity from ash is given in Aronsson and Ekelund (2004).

Radionuclide contamination

The Swedish Radiation Protection Institute has advised that no wood ash with activities exceeding 5 kBq kg^{-1} should be applied in forests (Ravila and Holm, 1996). In a north–south transect of Sweden, the background levels of radioactivity range from 0–40 kBq m–2 largely due to the Chernobyl fallout in 1986.

However, most of the activity is associated with soil rather than the vegetation (Thiry and Myttenaere, 1993; Högbom and Nohrstedt, 2001). A few studies have traced radioactivity through the lumber industry (Krosshavn *et al.*, 1996; Ravila and Holm, 1996), as it was expected that ashing would increase the concentration of radioactive elements such as radiocaesium and radiostrontium, but this was not found. Högbom and Nohrstedt (2001) added ash at moderate radioactivity levels (0–4 kBq kg^{-1}) to afforested field sites, but after 8 years there was no statistical difference in the tree composition between experimental sites and the controls. The activity of ^{137}Cs was still greater in the soils compared with tree components by a factor of three. At one site, however, radioactivity significantly decreased in all compartments, which Högbom and Nohrstedt (2001) attributed to the synergistic effect of K concentration in the ash with ^{137}Cs in the humus layer of the soil. Uptake of radioactive labelled substances into fruits and berries has been fully investigated by Levula *et al.* (2000).

Current evidence therefore suggests that any radioactivity added through spreading ash on forest floors is no greater than that naturally occurring from background fallout, and may decrease over time in the soil due to the K content of the ash. This finding is obviously dependent on there being no background rise of level and wood fuel not being taken from point sources of ionizing radiation or from geological regions with naturally high levels of radioactivity.

Soil water leachate

Studies on water chemistry from forest soils treated with wood ash in the USA, Sweden and

Table 6: Swedish Forestry Board recommended element concentration limits in wood ash for forest fertilization: Swedish National Board of Forestry, Draft Report (2002)

Elements	Standard values g kg⁻¹ DM		Elements	Standard values g kg⁻¹ DM	
	Lowest	Highest		Lowest	Highest
Calcium	125	–	Arsenic	–	0.300
Magnesium	20	–	Lead	–	0.300
Potassium	30	–	Cadmium	–	0.030
Phosphorous	10	–	Chromium	–	0.100
Boron	–	0.500	Mercury	–	0.003
Copper	–	0.400	Nickel	–	0.070
Zinc	1.000	7.000	Vanadium	–	0.070
Organic environmental toxins mg/kg DM			Total PAH		2

Here the standard values are given in $g\,kg^{-1}$ DM.

DM = dry matter.

Finland have been completed using a variety of surface water sampling (Fransman and Nihlgård, 1995; Nilsson and Lundin, 1996; Piirainen, 2001), soil suction methods (Ring *et al.*, 1999; Högbom *et al.*, 2001b; Van Hees *et al.*, 2003), lysimeters (Kahl *et al.*, 1996; Staples and van Rees, 2001; Saarsalmi *et al.*, 2005) and shallow wells (Williams *et al.*, 1996). Summaries of the ash application rates and results in these studies are given in Table 7.

Data from Kahl *et al.*, (1996) in Figure 3 summarize the nutrient change over time in the soil solution chemistry beneath white birch (*Betula papyrifera* Marsh.) and *Fagus grandiflora* beech woodland treated with ash. Minimal effects were observed at low ash dose rates, but heavier additions overloaded the buffering capacity. A rapid high concentration flush of all measured elements in the soil water was followed by a steady decline of values from a half or one-third of this peak concentration over the following 21 months for Ca, Mg, Cl, SO_4 and pH values. There was, however, no leaching of trace metals. Williams *et al.* (1996) showed in particular, increased concentrations of both Ca and K in the soils and groundwaters, with some movement of Mg and Al at relatively high ash loading rates from 11–44 t ha⁻¹. In this latter study, nitrates were also consistently higher on treated plots than on controls, peaking 12 weeks after the ash application. This was coincident with the highest recorded soil pH (6.25) in the experiment, under the heaviest ash loading. The occurrence of heavy metals showed no clear trends, but the low levels of Cd suggest that the Fe and Al oxides in the acid forest soils formed complexes with Cd, Zn and Mn. In the long-term experiment of Sarsaalmi *et al.* (2005) over a 10-year period, soil leachate at 20-cm depth taken from the mineral soil of a Haptic podzol showed increases of Ca, K and Mg, but no appreciable effect on NO_3 concentration, pH or Cd, Cr, Cu and Pb levels.

In Finland, powdered ash has traditionally been applied to forests in winter, onto the snow pack, which has been shown to increase the leaching loss of base cations and SO_4 (Piirainen, 2001). In contrast, summer applications reduced leachate loss. This work by Piirainen also monitored leaching of nutrients and heavy metals from two drained former peatlands (one *Carex*, one *Sphagnum* peat), now under P. *sylvestris*. The application of both fly ash and granulated bottom ash resulted in high concentrations of Ca, Mg and K base cations in the soil and groundwater, along with increased SO_4 concentration. Some rise in Cr was also detected at this site, but no increase of groundwater concentrations of Zn, Cd, Cu and Ni were found after any ash treatment at any site. In the 'Carex' site, with higher N content in the peat, increase in the NH_4 concentration was linked to possible N mineralization. Formation of NO_3 was found in soil leachate from depth at a southern Swedish site which has high atmospheric N deposition (Högbom *et al.*, 2001a).

Several authors monitored heightened values of NO_3 in the leachates from 'limed' soils (Geibe *et al.*, 2003; Lundström, 2003; van Hees *et al.*, 2003), but concluded that the effect was much reduced or undetectable at a comparable ash-treated site.

Table 7: A comparison of wood ash effects on soil leachate water

Area	Soil type	Ash loading t ha^{-1}	Effects monitored Soil	Water
Maine, USA (Williams *et al.*, 1996)	Loamy sands (unknown pH)	11–44	High surface K and Ca, no change at 45 cm. Concentration equalled control at 60 weeks	Small increase in Ca, K and SO$_4$. Heavy metals were below detection limits
Maine, USA (Kahl *et al.*, 1996)	Sandy spodosol, acidic pH 4	6–20	pH increased from 4–5, exchangeable Ca, K and Mg increased, Mn and Al decreased. Treatments declined to equal the control after 25 months	Transient increase in base cations, minimal response at low loads, raised concentration of H, K and SO$_4$ for 20 months, loss of N in solution at high ash loadings
Finland (Piirainen, 2001)	Drained bogs *Carex* and *Sphagnum*	3–6		All ashed sites increased in Ca, Mg, K and SO$_4$. Some sites increased in P, NH$_4$ and Cr
Finland (Saarsalmi *et al.*, 2005)	Haptic podzol	1, 2.5 and 5.0 (loose ash)	Increase of CEC in humus and increase of Ca, Mg and P. Increase of B, Cu, Fe, Mn and Zn positively correlated with ash dose (P = 0.001) and (P = 0.05) for Cd, Cr and Ni. No significant change in mineral layer, except in Ca, K and Mg	pH lower than control for 4/5 years, while Ca, Mg and exchangeable Al rose. Initial high K and SO$_4$ (S) fell to control levels after 6 years. Small increase in Mn and Zn. No increase in NO$_3$, Cd, Cr, Cu or Pb in percolated water
Sweden (Fransman and Nihlgård, 1995)	Podzol over granite	2.2 (fly ash)		Increased pH, Ca, K and Ca/Al ratios. Acidic leaching of NO$_3^{-1}$ decreasd Fe and Mn
Sweden (Ring *et al.*, 1999)	Podzol on till, clear cut and *P. sylvestris* forest	2 (6 products tested – loose to granulated)	After 3 years loose ash increased P, Cu and Zn on clear cut, none under pine. In trees pH raised 0.8 units and Al increased by pelleted ash	No significant ash treatment affects on pH – some increase in nitrate (N) from clear cut – none under pine
Sweden (Högbom *et al.*, 2001b)	Acidic till, high N deposition	4.2	Increase in Ca and Mg. pH rise of 0.2 units	Increase in NO$_3$

Wood ash effects on vegetation and soil biota

Forest fertilizer

Vance (1996) considered it more likely that hardwoods would benefit from ash application than softwoods due to their higher requirement for base cations (K, Ca and Mg). Hardwood nutrient ratios suggested for optimal growth are P : K : Ca : Mg at 1 : 5 : 20 : 2.5 and ash commonly delivers 1 : 7 : 45 : 2.5 (Vance, 1996). However, published research is dominated by work on conifers. Soil type is the principle factor determining tree responses, as it governs pH changes and thus the dynamics of nutrient release (Aronsson and Ekelund, 2004). Positive tree growth response to the use of wood ash has been widely reported
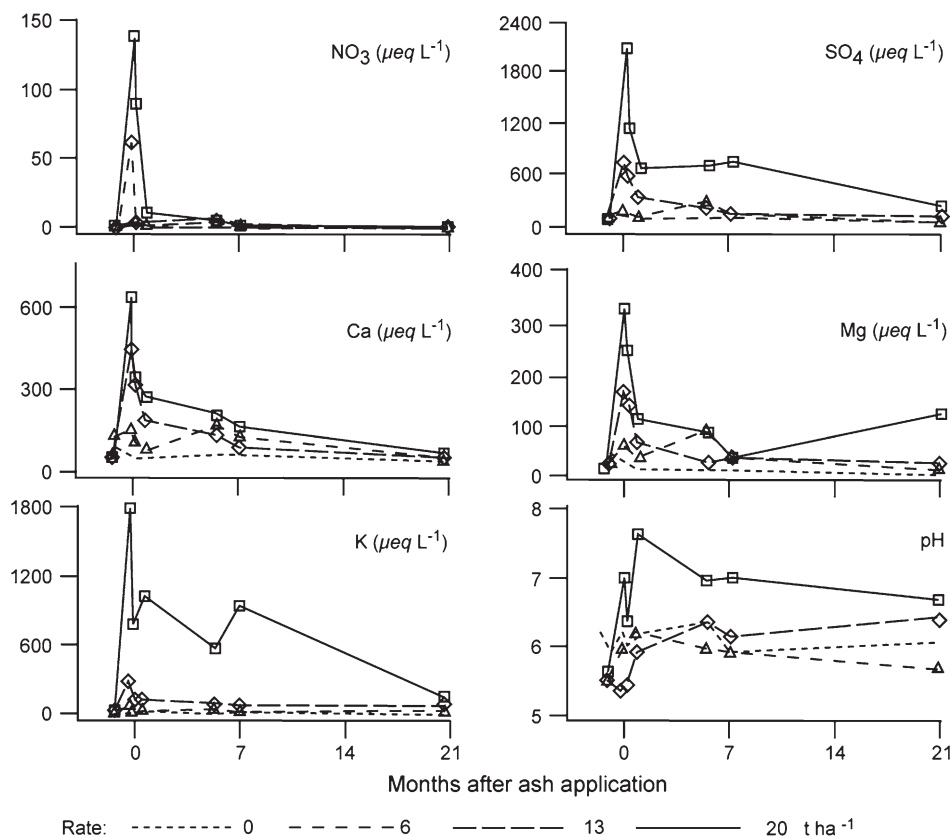
Figure 3: Changes in soil solution chemistry beneath white birch and beech woodland after ash treatment. Data are volume-weighted means of up to three replicate lysimeters in the same plot, as collection permitted. (Redrawn from Kahl *et al.*, 1996). NO_3 = Nitrate; SO_4 = Sulphate; Ca = Calcium; Mg = Magnesium; K = Potassium; pH = Units.

from studies on peat soil (Silfverberg and Hotanen, 1989; Ferm *et al.*, 1992) and ex-agricultural land (Etiégni *et al.*, 1991; Steponkus, 1992; Vance, 1996). Contraindications have also been published (Hüttl and Zöttl, 1993; Lundström *et al.*, 2003), although in most cases these studies investigated the combined effects of wood ash and lime (Eriksson *et al.*, 1998).

Observed effects of wood ash application on tree growth are reviewed here on the basis of soil type.

Peat bog and mire sites

Long-term tree growth experiments with wood ash spreading have been conducted for many decades in Fenno-Scandinavia as a result of the 'swailing' tradition, where tree planting is made into burnt forest clearings (Silfverberg and Moilanen, 2001). Initial studies evaluated the application of dry powder ash after snowmelt in spring. A sustained productivity increase of 3–4 m^3 ha^{-1} over 55 years was evident after wood ash applications of 5 t ha^{-1} in 1937 to drained peat (Silfverberg, 1996; Korpilhati *et al.*, 1999). The growth response was initially slower than that of commercial NPK fertilizers, but reached equivalence at 10 years. Moilanen *et al.* (2002) estimated the increase in wood production over 1947–94 on an ash-treated drained mire site to be between 13 and 17 times that of the control plot. Soil nitrogen in the low-dose ashed plots increased in the 0 to 20 cm soil layer by 18 times and soil

phosphorous by 9 times, accompanied by notable cellulose decomposition of the litter, vegetation and peat. The most dramatic effect of wood ash application on tree growth was reported by Ferm *et al*. (1992) who reported a standing volume increased from 15 to 70 m^3 ha^{-1} at the highest rates of application (5 and 10 t ha^{-1}). They also recorded peat decomposition, indicating enhanced N mineralization under the ash.

In short-term experiments using peat substrate, wood ash has been shown to enhance the growth of seedlings at low application rates, but both the germination and initial growth of tree seedlings are reduced at application rates over 5 t ha^{-1} (Heikinheimo (1915) cited in Rikala and Jozefek, 1990). Seedling root extension is encouraged by ash additions (Persson and Ahlstrom, 1994), and Clemensson-Lindell and Persson (1995) have shown greater fine root length extension in wood ash-treated soils compared with ammonium sulphate fertilization. It is also apparent that the nitrogen status of the soil determines the response of young trees to wood ash addition. Weber *et al*. (1985) established *Salix* saplings and *Alnus incana* on an N-rich mire site previously cultivated 16 years ago, and compared productivity in wood ash treatments of 10 t ha^{-1} with NPK fertilizer added at a rate of 150 kg ha^{-1} N, 92.4 kg ha^{-1} P and 382 kg ha^{-1} K. Wood ash increased biomass production by 65–70 per cent after 1 year's growth. Significant soil N mineralization in the soil had occurred, as indicated by a rise in the soil C : N ratio. Compared with conventional fertilizer treatment, wood ash had less effect on soil denitrification, and less depletion of water-soluble organic matter. On N-limited sites, however, wood ash application at 5–29 t ha^{-1} had no fertilization effect on willow and no enhancement of N mineralization, although K and P levels in the soil were higher (Lumme and Laiho, 1988).

In a two-season nursery trial growing *Acer rubra* seedlings in the humic O-horizon of a sandy loam, Unger and Fernandez (1989) found that soil pH in this soil was raised 0.59 units and foliar levels of K and Na in the seedlings rose significantly over seedlings grown in B-horizon soil.

Nutritional and tree health surveys have also reported beneficial effects of wood ash to forest trees. Moilanen *et al*. (2002) reported fewer symptoms of tree disorders and decreased levels of dieback, even at high doses of ash, with increases in foliar levels of P and K 40 years after application at both 8 and 16 t ha^{-1} sites, compared with the deficit control treatment. No plant nutrient deficiencies were found on plots treated with 16 t ha^{-1} of ash, but some P and K deficit was found with lower additions and trees in the untreated plots showed deficiency throughout the study.

Podzolic soils

Korpilahti *et al*. suggested in 1999 that growth response to ash addition is more successful where there is already high N in the soil profile (1.5–2.5 per cent dry weight). Where the N content is low (<1 per cent), wood ash is unlikely to have an effect directly on growth. This has been confirmed by Jacobson (2003) who monitored stem growth of Scots pine and Norway spruce at four locations across Sweden on podzolic soils, where granulated wood ash was applied at three dose rates to a maximum of 6 t ha^{-1}, with and without additional N fertilization. In all three southern experiments tree growth was variable, but generally increased in ash-treated soils over the 11 years (4–10 per cent increase in wood volume), to a maximum of 1.2 m^3 ha^{-1} y^{-1} production volume for each addition of 1 t ha^{-1} wood ash applied. However, at the northern site (64° N) the opposite effect was observed, with growth reduced by 0.8 m^3 ha^{-1} y^{-1} compared with control treatments. The authors conclude that the mobilization of N is critical, and that poorer soils and more severe climatic conditions of north Sweden led to N immobilization in mor soils. Soils in the south are known to be N enriched by pollution (Lövblad *et al*., 1995). The C : N ratio has been suggested as a good indicator of this potential effect in response to wood ash application, with the threshold value of 30 in the humic layer indicating the likelihood of N mobilization after ashing (Persson, 1988). In the most recent draft recommendations for ash fertilization drawn up by the Swedish National Board of Forestry (2002) suggested ash doses are varied according to latitude to minimize this effect (Table 8). A similar long-term study across Finland by Saarsalmi *et al*. (2004) monitored tree growth in five coniferous stands at time intervals of 5 and 10 years after treatment, after each had been given ash doses of 3 t ha^{-1} and three given additional N fertilization. The substrates were sandy

Table 8: Swedish maximum values for input of heavy metals from ash to forest land by region (Swedish National Board of Forestry, 2002)

Heavy metal	South Sweden (S of 60°)		Central Sweden Loading rate g ha^{-1}		North Sweden (N of 64°)	
	Spruce*	Pine	Spruce	Pine	Spruce	Pine
As	90	30	90	30	90	30
Cd	100	100	50	50	25	25
Cr	300	300	200	200	150	200
Cu	1200	800	600	800	500	500
Hg	10	5	10	5	10	5
Ni	200	200	200	200	200	200
Pb	1000	400	500	200	250	100
Zn	20 000	10 000	15 000	7000	10 000	5000
Va	200	100	100	50	100	50

*NB values for spruce (deciduous) are also those suggested for broadleaf trees (other data not available).

till soils, in which pH increased by 1–1.7 units in the humus layers, but only 0.3–0.4 in the mineral layers beneath. Levels of total and extractable calcium and magnesium rose in the humus layer with wood ash addition, but no significant tree volume increase was detected. However, tree growth in plots with both ash and N matched that in plots given a traditional site-specific fertilizing treatment. The latter study did not find increases in the nutrient status of the foliage, but Arvidsson and Lundkvist (2002) have recorded positive nutritional changes in sites across four different climatic zones in Sweden, which included podzolic soils under *Vaccinium*. Hardened wood ash at 3 t ha^{-1} had been applied to whole-tree harvested plots replanted with young Norway spruce, where after 5 years they noted an increase in P, K and Ca concentration in the needles at all sites, though no significant increase in the tree growth. Bramryd and Fransman (1995) reported similar findings from long-term experiments on Scots pine growing on a ferric podzol.

Loams and other soil types

A limited number of experiments investigating wood ash addition to loam soils indicate that the highest increases in tree growth are achieved at low dose rates over long time periods (Etiégni *et al.*, 1991; Vance, 1996). Unger and Fernandez (1989) using the B-horizon soil from a Hermon sandy loam, found no significant increase in growth rate of *Acer rubra* seedlings over the controls with a range of wood ash addition from 4–20 t ha^{-1} after 18 weeks. pH increases were less than in O-horizon soils, and were attributed by the authors to cation exchange equilibria. Extractable Al was significantly greater in the soils when N was added, and increased foliage concentrations resulted – the authors concluded that the lack of seedling growth response to ash addition might be the short length of the experiment or the young age of the seedlings. One other American study growing short rotation coppice in New York State on a Palmyra gravely loam has reported similar non-significant response of growth from existing *Salix purpurea* stools, given ash doses between 0 and 20 kg ha^{-1} annually for 3 years (Park *et al.*, 2005). Although stem size increased, stem numbers declined, so that biomass production remained static or even decreased over the 3 years. However, soil pH increased in all treatments and extractable levels of K, P, Ca and Mg also rose. The authors concluded that additional sources of N are required with the ash to maintain the productivity of the stand.

Spruce seedling growth under soil amendment with paper mill sludge and ash mixtures have been trialled by Shepard (1995) on *Picea mariana* and in Canada on *P. glauca* by Staples and van Rees (2001). Both studies varied application rates between 0 and 5 t ha^{-1}. In the first study, soil pH

was raised from 4.8 to 6.9 units and a decrease in spruce growth recorded at higher dosage rates was attributed to salt phytotoxicity effects of the ash from salt laden wood. In the second study, forward growth of second and third year seedlings was inhibited by weed competition.

Effects on ground vegetation and soil biota

Higher plants

Early papers on forest understorey vegetation concentrated on investigation of the heavy metal concentrations in berries harvested for human consumption, particularly from *Vaccinium* species (Silfverberg and Issakainen, 1991). More recent studies by Levula *et al.* (2000) and Perkiömäki *et al.* (2003) have confirmed the findings that no increase in cadmium or lead content of the berries occurs, even when high Cd content ash has been applied and peaty soil pH increased. On mineral soils berry harvest increased following ash application, but neither Cd nor Pb concentrations were elevated above normal (usually <1 mg kg^{-1}). *V. vitis-idaea* berries were examined by Levula *et al.* (2000) for heavy metals, S and ^{137}Cs concentrations after the application of wood ash, and ^{137}Cs was decreased in response to wood ash from conifer bark. In contrast, higher concentrations of Cd have been reported in edible fungi such as ceps (*Boletus edulis*) after ashing (Rühling, 1996) and *Lactarius* species after Cd spiked ash treatment (Perkiömäki *et al.*, 2003).

In terms of ground vegetation change, a small reduction in the dominance of *V. vitis-idaea* was noted by Levula *et al.* (2000) in response to wood ash application of 5 t ha^{-1}, but this conflicts with the results of Gyllin and Kruuse (1996) who found no difference in *Vaccinium* stands up to 9 years after wood ash application. An overall decrease in dwarf shrubs cover has been recorded by Jacobson and Gustafsson (2001) in ash-treated soils under Scots pine in Central Sweden, and Silfverberg and Hotanen (1989) recorded changes in a 'drained' herb-rich peatland treated with ash, with N-demanding species such as *Cirsium helenoides*, *Daphne mezereum*, *Paris quadrifolia*, *Prunus padus* and *Urtica dioica* becoming more dominant over time. Heather (*Calluna vulgaris*) has been shown to be suppressed by grasses such as *Deschampsia flexuosa* after wood ash application (Arvidsson *et al.*, 2001, 2002), although the bilberry cover was unaffected.

Bryophytes and lichens

In a review of the effects of wood ash application on lower plants, Kellner and Weibull (1998) concluded from earlier Scandinavian research (Jäppinen and Hotanen, 1990) that most moss species survived well with ash dosage rates of under 7 t h^{-1}. Effects on certain species such as *Goodyera repens* (Kellner, 1993) and *Peltigera aphthosa* (Nohrstedt *et al.*, 1988) were however evident. Where wood ash led to browning and death of some of the leaf shoots in the first 6 months after application and cover was reduced after 1 year, complete recovery of the moss flora was evident after 5–10 years. However, the balance of the moss species had changed within the sward, with the proportion of *Pleurozium schreberi* increasing at the expense of *Dicranum polysetum* and *Ptilium crista-castrensis*. Jacobson and Gustafsson (2001) monitored significant losses of moss cover after 5 years only in the highest ash treatments (9 t ha^{-1}) – at lower dose rates, there was no discernable visible damage to the bryophytes, though reductions of cover in the lichen genus *Cladina* were recorded. Wood ash effects depend upon the surface area of the moss in direct contact with the particles – large granulated particles have less area contact than loose ash, and the use of this form to lessen the impact is recommended (Gyllin and Kruuse, 1996). Where Bryophyte communities are important elements of the ecosystem, application rates of <2 t ha^{-1} are advised (Kellner and Weibull, 1998).

Soil biology

Very few published data exist on ash effects of wood ash on soil fauna except for studies on enchytraeid and earthworm populations by Lundkvist *et al.* (1998), in which Cd levels in body tissue of worms were monitored from forest plots with low ash application levels (3.2 t ha^{-1} of granulated ash). Cd levels increased in the body tissue in the second year after ash application, but no adverse effects on the population was recorded.

In contrast to soil fauna, studies investigating the changes in soil 'microbial' activity under ash application are more numerous from Finland, Sweden and Switzerland (Karsisto, 1979; Fritze, 2001). Soil pH was proposed as the principal driver of an observed increase in N mobilization and cellulose decomposition rates in response to wood ash application to peat (Weber *et al.*, 1985). Large increases in bacterial populations have been reported by Bååth and Arnebrant (1994), and soil respiration was always higher in the ash amended treatments, but only sustained where the soil C : N ratio was high. Ash has also been shown to increase respiration without change to soil fungal biomass (Fritze *et al.*, 1994) and with increased biomass (Zimmermann and Frey, 2002) – the latter was attributed to a higher pH rise and increased mineralization of organic matter. Aronsson and Ekelund (2004) have reviewed the available data for effects on soil decomposer animals and aquatic systems.

Soil fungi

EM fungal communities have been shown to respond to wood ash application, with an increase particularly in *Cortinarius* sp (Erland, 2001). In this study, ash granules were heavily colonized by four particular mycorrhizal fungal types, three of which were also found on the tree roots. Further analysis of this dataset revealed that these species together made up to 55 per cent of the total EM fungal community on the screened roots (Mahmood *et al.*, 2002). Van Breemen *et al.* (2000) suggest an active role for these EM fungal types in mobilizing nutrients from weathered ash on the forest floor. It seems likely that wood ash addition also increases the ability of EM fungi, such as *Piloderma*, to colonize tree roots. This contrasts with the negative effects of lime application on EM species diversity, as reported by Taylor and Finlay (2003), although their study found no beneficial effects of wood ash treatments.

Conclusions

Key findings on the potential use of wood ash as a soil amendment in forestry resulting from this review are summarized in each first paragraph of the synthesis. Comments on these findings related more specifically to the potential use of wood ash in British forestry appear in the accompanying second paragraph.

Synthesis of current research

1 The quality of the wood ash is dependent on the raw material burnt, and data are available for ash deriving from burns of single species of trees which have a different composition to that from mixed wood fuels with and without bark. Ash from hardwood trees is likely to be a more effective fertilizer than that from softwoods, as it contains higher proportions of macronutrients. Softwoods frequently have high levels of silica trapped in their bark during harvesting from sandy sites, which can have a detrimental effect on the granulation of the material after burning.
   - There is no published data of ash quality derived from two of the most common British timber trees that might be used for biomass burning – Sitka spruce from the uplands and north and Corsican pine from the lowlands. There is also none yet for short rotation coppice species, based on willow, poplar or sweet chestnut, or for short rotation forestry that might be based on sycamore, ash and hornbeam, or introduced species such as eucalyptus.
2 The effects of burn processes in commercial furnaces have been well researched, and temperatures need to be between 6 and 900°C to release the maximum amount of nutrients without creating heavy metal volatilization . The fly ash fraction should not be included in any application of ash to the land.
   - The effects of co-firing, currently being undertaken in some British power generation with non-wood products, make assurance of the ash quality more complex – in particular prediction of the heavy metal content. Frequent routine chemical analysis will be needed to check quality.
3 The physical form of the ash is most important to controlling delivery of elements to the forest. The granulated form has been shown to be more easily spread, with slower release of chemical elements which reduces the risk of high magnitude alkaline flushes to the natural environment.

- The technology of wood ash granulation has been perfected in Scandinavia, and any wood ash use on a commercial scale will need to invest in similar systems.

4 The rate and order of nutrient release from wood ash under laboratory conditions are well established, particularly from American research – an absence of N in the product is normal.

- The release of nutrients in the field is subject to many other variables on a site-to-site basis – these include rainfall and temperature regime, as well as soil and surface litter factors.

5 The fertilization effects of wood ash have been shown to have equivalence with commercial NPK compounds, particularly when used on ex-arable and organic forest soils. Its neutralizing capacity relative to lime has also been established. In tree crop systems, wood ash could thus be applied once a rotation to conifers, at the point of most rapid growth in the thicket or early mid-rotation stage.

- The high pH of the raw material limits its use on shallow alkaline soils, such as those found on steep slopes developed on chalk or limestone geology in southern Britain. Its effects on very free draining sandy soils in the lowlands might be similar to those recorded on glacial deposits in northern Europe.

6 Additions of wood ash from known wood sources to forest soils should not increase the heavy metal loadings on sites beyond any of those currently recommended by British or European environmental authorities. Wood ash has similar or lower concentrations to commercial fertilizers.

- Due to the wide range of soil types in Britain, and the possibility of planting in land contaminated from past industry or commerce (Brown Field sites), dose rates will need to be adjusted to the receiving area soil type as well as tree canopy type.

7 Dose rates of 10 t ha^{-1} have been suggested by Vance (1996) as being sufficient to replace all nutrients lost by whole-tree harvesting – but ashing rates over 5 t ha^{-1} have been shown to have deleterious effects on ground flora in upland areas. Decline of woody shrubs, including *Calluna* and *Vaccinium* have been reported along with major changes to Bryophyte and Lichen communities.

- In British upland conifer plantations, which have been shown to be particularly rich in bryophytes and fungi (Humphrey *et al.*, 2000, 2002), high ash doses could be very detrimental to the overall biodiversity of the stands. Atlantic oakwoods, which have specialist epiphytic communities and the lowest recommended N critical loads levels in forestry, would also be at risk of major change. Experimental applications of low ash dose rates (<2.5 t ha^{-1}) are needed in both situations.

8 Wood ash addition to organic or peaty forest soils has been shown to have a long-lasting effect. Due to raised pH, organic matter breakdown rates are accelerated and N is released for take up by the trees. However, very acid soils should be avoided as the increased pH may also subsequently mobilize Al and Fe with leaching into the subsoil.

- Peat and mire mineralization results in eventual surface layer disintegration – compared to Scandinavian forestry, Britain has far less of this resource available to afforest, and restoration of wet land sites, and bog woodland in particular, now takes precedence over timber production (EU Habitats Directive category 91D0, 1992).

9 Swedish research has shown that wood ash use should be reduced on sites that are heavily enriched by N air pollution, as this increases the likelihood of N leaching into groundwater and surface streams. Their recommended ash loadings vary according to latitude to take into account the calculated Critical N Load, which has a mean value of 8 kg N ha^{-1} yr^{-1} in the south and a maximum of 12 kg ha^{-1} yr^{-1}.

- In Britain, the recommended limit of N critical load has also been set at 12 kg N ha^{-1} yr^{-1}, but this calculated load has already been exceeded in many forest areas for both managed and unmanaged, conifer and broadleaf stands (Hall *et al.*, 2004). Additional nitrogen release by ash application to upland soils, where total N deposition has been greatest between 1900 and 2000 (Fowler *et al.*, 2004), should be avoided. However, spatial variability of N deposition which has been mapped at 5 km^2 resolution may

contribute to an overestimate of the proportion of UK woodland affected by excess deposition. A conceptual model of spatial variations in N leaching based on an upland catchment in Wales developed by Evans *et al*. (2004) identified the most likely 'nitrate leaching zone' according to hydrology, topography and soil types. In order to use ash fertilization as a useful soil amendment in upland catchments, such as after whole-tree harvesting (WTH), it will be necessary to identify areas least susceptible to N leaching using this kind of geographical approach, on a site by site basis.

10 It has been shown in a small number of published studies that ash addition to loam soils has a limited effect on woody biomass production, unless there is additional N available. Although rises of extractable P, K, Ca and Mg do occur, pH rise is small. To date there is no published research on wood ash effects on clay soils.

- In Britain, it has been proposed that production of biomass tree crops might be encouraged in lowland situations, particularly on ex-arable land. The use of the wood ash derived from burning these crops, back to the harvested sites as fertilizer, would be a sensible cyclic system operated within economic transport distances of each power plant. In lowland Britain, many soils will be basically brown earths, with/without podzolization if derived from sandstones or gleying if derived from clay geology. Some 32 lowland areas have been designated as 'nitrogen sensitive' (DefraUK, 2006) due to excessive pollution by N from agricultural practices in the past on land over sensitive geology such as chalk aquifers. Farming systems are closely monitored for N enriching procedures, and until basic research is completed on the effects of wood ash addition to such brown earth soils, it is likely that wood ash fertilization will be restricted in these areas.

Concluding remarks on the environmental effects of wood ash application to forest land

This review has shown that ash treatments at low levels have been successfully used in both the USA and Europe for nutrient replacement into poor forest soils. Nutrient losses from WTH are known to be up to three times those from conventional stemwood harvesting. Repeated whole-tree harvesting in the eastern USA has been estimated to remove 20–60 per cent of total site Ca, and 2–10 per cent of K, P and Mg over a 120-year period (Federer *et al*., 1989; Vance, 1996). Swedish research by Arvidsson and Lundkvist (2002) and British research in upland Wales by Stevens *et al*. (1995) have reported similar potential nutrient losses. There are also examples where wood ash amendment has been proposed to correct specific soil deficiencies, as suggested in the research, such as mitigating the K and Mg deficit in soils of the NE and southern Canada, or P levels in soils under pine forests in the south USA.

Environmental problems associated with wood ash use are less likely to come from its heavy metal content, which can be partly removed at source during the burn, and more likely to be associated with its high Ca content. This may create a rise in soil pH, increasing microbial populations and the potential mobilization of N. Although the rise in pH is potentially beneficial to tree growth on acidic or nutrient poor soils, it can be accompanied by changes to the ecology and functioning of forest ecosystems. Individual site characteristics and the nature of the ash should govern decisions on the use of the material. The importance of the receiving soil type becomes evident from both short- and long-term experiments in the field.

Appendix

Danish Legislation relating to wood ash application (2001) (SkogForsk Report No. 2, 2001)

1 No recycling of wood ash with Cd content over 15 p.p.m. Specifically no fly ash. Three classes of content are designated according to Cd level, of which a content of 0.5–8.0 p.p.m. is the mid point.

2 Other heavy metal contents are restricted to Pb: 120 p.p.m., Hg: 0.8 p.p.m., Ni: 30–60 p. p.m., Cr: 100 p.p.m.

3 P application shall be limited to a maximum of 30 kg ha^{-1} or total of 90 kg ha^{-1} over 3 years.

4 No more than 7.5 t of wood ash may be applied to a stand over a single rotation, with a lower limit if the Cd content is in the upper band.

5 Analysis for polyaromatic hydrocarbons (PAH) must be made where residual carbon content is over 5 per cent. PAH content must not exceed 3 mg kg^{-1}.

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